Exploring the consequences of climate change for indoor air quality*

William W Nazaroff

Department of Civil and Environmental Engineering, University of California, Berkeley,
CA 94720-1710, USA

E-mail: nazaroff@ce.berkeley.edu

Abstract
Climate change will affect the concentrations of air pollutants in buildings. The resulting shifts
in human exposure may influence public health. Changes can be anticipated because of altered
outdoor pollution and also owing to changes in buildings effected in response to changing
climate. Three classes of factors govern indoor pollutant levels in occupied spaces:
(a) properties of pollutants; (b) building factors, such as the ventilation rate; and (c) occupant
behavior. Diversity of indoor conditions influences the public health significance of climate
change. Potentially vulnerable subpopulations include not only the young and the infirm but
also those who lack resources to respond effectively to changing conditions. Indoor air
pollutant levels reflect the sum of contributions from indoor sources and from outdoor
pollutants that enter with ventilation air. Pollutant classes with important indoor sources
include the byproducts of combustion, radon, and volatile and semivolatile organic
compounds. Outdoor pollutants of special concern include particulate matter and ozone. To
ensure good indoor air quality it is important first to avoid high indoor emission rates for all
pollutants and second to ensure adequate ventilation. A third factor is the use of air filtration or
air cleaning to achieve further improvements where warranted.

1. Introduction

Indoor air pollution can affect public health. Most of the air that humans encounter on a daily basis is indoor air. Many pollutants that are potentially health hazardous are emitted from indoor sources. Such emissions occur from building materials, from products used or stored indoors, and from processes that occur in indoor environments. Because of contributions from indoor sources, indoor levels are higher than those found outdoors for many air pollutants in many buildings. In addition to contributions from indoor sources, pollutants intrude into buildings from outdoor air along with ventilation. For some pollutants that are predominantly of outdoor origin, buildings offer some protection, albeit incomplete. Also, some outdoor pollutants that enter a building interact with its components or contents, thereby altering the composition of indoor air in ways that can have health and welfare consequences for occupants.

Indoor environmental quality has many facets. This letter focuses on the chemical and particulate pollutants. Specifically, this paper is concerned with volatile and semivolatile molecular pollutants, both organic and inorganic, and also with suspended particulate matter. In the case of particles, abiotic materials are emphasized, although a brief discussion of allergens associated with pollen is included. Other indoor environmental quality concerns that

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may be associated with climate change are not addressed in
this paper, such as challenges for maintaining appropriate
thermal conditions indoors (de Wilde and Tian 2012) and
managing the risks from more frequent or more extensive
flooding (Taylor *et al* 2011). Also not discussed here are
biological concerns associated with microbial agents, insects
and arthropods, and mammals, along with concerns that arise
because of efforts to control these agents. These matters
are addressed in the full report of an Institute of Medicine
committee, Climate Change, The Indoor Environment, and
Health (IOM 2011). Spengler (2012) provides a brief
summary of the full report. Further restricting the scope:
this paper emphasizes conditions in countries with advanced
economies, with much of the reviewed literature focusing
specifically on the United States. Major public health
concerns associated with the use of solid fuels for cooking,
as is prevalent in the rural parts of developing countries,
are not addressed here. Nor does this letter consider the
indoor air quality aspects associated with the challenging
ambient air quality problems in the increasing number of
global megacities in developing countries (Gurjar *et al* 2008,
Chan and Yao 2008).

With regard to the issues that are addressed in this article,
there is little published literature that specifically considers
the effects of climate change on indoor air quality that
would influence public health. However, substantial research
has been published on many components of this system.
For example, there is a substantial emerging literature on
the impact of climate change on air quality from urban to
global scales (Jacob and Winner 2009), with specific attention
to climate-change effects on particulate matter (Tagaris
et al 2007) and ozone (Bell *et al* 2007, Racheria and Adams
2009), and on related health effects (Tagaris *et al* 2009).
A voluminous literature characterizes health risks associated
with pollutants in outdoor air (Dockery *et al* 1993, Bell
et al 2004, Pope and Dockery 2006, Jerrett *et al* 2009,
Pope *et al* 2009, Sun *et al* 2010). Considerable research
documents our understanding of indoor–outdoor relationships
for important air pollutants, including particles and ozone
(Wallace 1996, Weschler 2000, Monn 2001, Jia *et al* 2008).
Research has explored the extent to which health risks
associated with outdoor pollution are a consequence of indoor
exposures (Wilson and Suh 1997, Wilson *et al* 2000, Weschler
2006, Bell *et al* 2009, Chen *et al* 2012). There is also a large
body of work reporting on the ways in which sources of indoor
pollution influence indoor air quality and human health (Jones
1999, WHO 2010). Logue *et al* (2012) have estimated that
air pollutant exposures in US residences, excluding exposures
to secondhand tobacco smoke and radon, contribute a loss of
400–1100 disability adjusted life years (DALYs) per 100 000
persons yr^{-1}. They found that fine particulate matter ($PM_{2.5}$),
acrolein, and formaldehyde 'accounted for the vast majority
of the DALY losses caused by (indoor air pollutants) $\cdots$
with impacts on par or greater than estimates for secondhand
tobacco smoke and radon'.

Apart from the few following examples, there is little
published research that links climate change to changes in
the levels of indoor air pollutants that might then influence

public health. Ayres *et al* (2009), summarizing how climate
change is expected to affect respiratory health, called for
more research on 'the role of housing and indoor climate
control systems in respiratory diseases'. Bell *et al* (2009) used
an epidemiologic approach to discern that communities with
higher air-conditioner prevalence had lower health effects
associated with outdoor levels of particulate matter. The use
of air conditioning for residential climate control would be
expected to provide better protection against outdoor particles
than opening windows. Peden and Reed (2010) review the
many ways that indoor and outdoor pollution influence the
prevalence and severity of allergic diseases. They discuss the
role that climate change will have in altering the spatial and
temporal patterns of outdoor aeroallergens. Wilkinson *et al*
(2009) evaluated co-benefits for mitigating climate change
and improving public health that would result from improving
the residential building stock in the UK (and also from an
improved stoves program in India).

Even though the aggregate system has not yet been
well studied, elements are sufficiently well understood to
permit defining a broad framework of the issues that need
to be considered and to illustrate specific aspects of potential
interest and concern. That approach is taken here. The paper
first outlines a framework for understanding how exposure to
pollutants in indoor air can be related to important influencing
factors. The paper then explores how climate change might
influence those factors. The approach relies heavily on
inference and so is constrained to consider portions of the
system that are well understood mechanistically. Available
evidence extrapolated to explore an unknown future is more
reliable when inferences are based on a cause-and-effect
understanding of the system rather than relying on studies that
reveal associations without providing clear evidence about
processes. Considering these limitations, this paper stresses
how climate-change phenomena might induce changes in
pollutant concentrations and exposures indoors. In a few
cases, the mechanistic level of understanding is sufficient
to relate anticipated changes in future exposures to health
consequences.

2. Framing the issues

2.1. Factors governing indoor pollutant concentrations and exposures

Fundamentally, exposures occur when people and pollutants
intersect in space and time. For indoor air pollutants,
exposures depend on concentrations during periods of
occupancy. Three classes of factors govern pollutant
concentrations in occupied indoor environments. One
category pertains to pollutants and includes attributes such
as the outdoor air concentration, whether the species is
gaseous or particulate, and pollutant dynamic properties such
as reactivity for gaseous species and the size of particles.
A second category of factors pertains to buildings and
includes the air-exchange rate, the presence and effectiveness
of deliberate air cleaning processes, and the types and
conditions of materials that comprise the building surfaces

and furnishings. This category would also include factors that affect pollutant emissions from materials associated with the building and its (non-human) contents. A third category of factors pertains to occupants and includes the timing of their presence indoors, occupant density, and activities undertaken that may influence both pollutant emissions and exposure. Each of the three categories of factors is richly complex: pollutants, buildings, and people are both numerous and diverse with regard to many attributes. Each of these categories of factors can also strongly influence indoor environmental quality and its public health consequences.

It is convenient to decompose indoor air pollutant levels into two parts: (a) indoor levels that result from indoor emission sources; and (b) indoor levels that result from the penetration and persistence of outdoor pollutants. For many pollutants, these two components do not directly interact and so the total indoor concentration burden can be represented as their arithmetic sum. That idea is built into many mathematical models of indoor pollutant levels. Ott *et al* (2000) have illustrated such decomposition for the specific case of airborne particulate matter (PM_{10}).

The air-exchange rate of a building can substantially influence indoor air pollutant concentrations. For pollutants of outdoor origin, ventilation serves as the means by which the pollutants are introduced into an indoor environment. Also, whether pollutants are of outdoor or indoor origin, ventilation is an important removal mechanism that limits pollutant accumulation indoors. In fact, a main purpose for ventilating buildings is to remove indoor-generated pollutants, including the metabolic gases emitted by human occupants. In general, higher ventilation rates cause indoor air to become more like the local outdoor air. Conversely, as ventilation rates are reduced, indoor air becomes progressively less influenced by pollutants of outdoor origin while becoming more strongly influenced by indoor sources.

2.2. How climate change could influence indoor pollutant levels

Climate change could influence indoor air pollutant levels in many ways. First, considering the existing building stock, significant influence can be anticipated from these causes, among others. (a) The levels of outdoor air pollutants change, which would affect human exposure indoors to pollutants of outdoor origin. (b) Buildings are operated in a different manner, for example with respect to ventilation rate or air-conditioner use, and these changes would alter indoor pollutant concentrations. (c) Humans adjust or alter their activities as a result of climate change, for example changing where they spend time or what they do indoors, and these changes would alter both the levels of indoor pollutants and the resulting exposures.

Because climate-change impacts are expected to occur over decades, one also should anticipate concomitant changes in the building stock that could significantly influence the nature and impact of climate change on indoor environmental quality and public health. There might also be changes in the ways humans behave in buildings that evolve over decadal time scales and materially alter indoor pollutant concentrations and exposures.

2.3. Adaptation, mitigation, and rapid responsive behaviors

A change in building design, building operation or habitual indoor human behavior that is influenced by climate change might be categorized as either an 'adaptation' or 'mitigation'. An adaptation represents a response to climate change aimed at providing protection against its impacts. Increased use of air conditioning would be an adaptation in response to higher average ambient temperatures. Mitigation is a change made to reduce or offset the anthropogenic impact on climate. Large proportions of society's use of fossil fuels serve to meet needs associated with buildings. Buildings are already and will likely continue to be settings where improved energy performance is sought. Some changes motivated by the goal of saving energy can have consequences for indoor environmental quality and public health.

Beyond anticipating adaptation and mitigation changes, one should also be mindful of responsive behaviors to climate catastrophes that may themselves have significant consequences for indoor environmental quality and public health. Examples would be actions taken to protect people and property in response to floods, wild land fires, extreme heat events, or power outages. As one very specific concern, discussed in more detail later, use of portable electricity generators following major storms has been associated with carbon monoxide poisonings (Hampson and Stock 2006).

2.4. Diversity of indoor environments

The impact of climate change on indoor air quality will vary with building type. The consequences of these changes will depend on how long people spend in different types of indoor environments and also on differences in the populations that occupy buildings. People spend most of their time in their own residences. For example, Jenkins *et al* (1992) reported that Californians spent an average of 62% of their time in their own homes and an additional 4% of their time in others' homes. Other important nonindustrial indoor sites reported for Californians were 'office building, bank' (5%), 'school' (3%), 'shopping mall' (2%), 'restaurant' (2%), and 'hospital/Dr's office' (1%). Klepeis *et al* (2001) reported analogous findings based on a US nationwide survey.

Children spend a high proportion of their time in school and they are considered more vulnerable than adults to adverse health effects from air pollution. Hospitals and other indoor environments where health care is provided would be of special concern because those who are already ill are more vulnerable than those who are healthy to further health insults.

Differentiating among building types is important for reasons that extend beyond the different populations that inhabit them. Different classes of buildings may be designed, operated, and maintained differently in ways that affect responsiveness to climate change. As one example, office buildings in the United States are commonly mechanically ventilated whereas the existing stock of residential buildings is mainly ventilated by a combination of air leakage (infiltration) and natural ventilation (through open windows or doors). Mechanical ventilation of residential buildings

will become more common in the future owing to recent changes in ventilation standards (Russell *et al* 2007). Other important changes have occurred in the building stock over time, such as the increased prevalence of air conditioning in US residences (Weschler 2009). Data from 2009 indicate that 87% of US residences are equipped with some form of mechanical air conditioning, up from 68% in 1993 (www.eia.gov/consumption/residential/reports/2009/air-conditioning.cfm).

Buildings also differ in the types of pollutant-emitting sources of concern. Cooking is a dominant activity in restaurants, common in residences, and rare in offices. The responsibility for maintaining appropriate environmental conditions in buildings varies among building classes and this variability influences the appropriateness of various policy options to address the public health concerns discussed here.

Indoor environmental quality conditions have the potential to be more broadly variable than corresponding outdoor conditions. In the United States, more than half of the population lives in the 52 most populous metropolitan statistical areas (MSAs). While there is some local and neighborhood variability in air pollutant levels within these areas, there are also some common characteristics such that the air quality of each MSA can be reasonably characterized using a small number of monitoring stations. Furthermore, actions taken by individuals within these MSAs have muted influence on urban air quality. By contrast, the population of the US resides in approximately 100 million individual residential units and there are tens of millions of other occupied buildings. What happens in individual buildings strongly influences the air quality within those buildings but does not significantly influence air quality in other buildings. In turn, the air quality within a single building can affect the health of people occupying that building, but generally would not impact others.

Diversity in the building stock is especially important for understanding the public health significance of how climate change might impact indoor environmental quality. Subpopulations that are potentially vulnerable to climate-change-induced impacts on indoor environmental quality include not only those who may be more susceptible to air pollutant health effects because they are young, old, or infirm. Subpopulations of concern also include those who may lack the financial resources or the appropriate knowledge to act wisely in response to an emergency induced by a climate-change event.

2.5. Ventilation time scales

Allowing for this broad diversity, what then are the factors that affect indoor pollutant concentrations? Applying the principle of material balance, the concentration of a given pollutant in any particular building can be determined by accounting for the net effect of source terms and removal processes. Sources include outdoor air and direct indoor emissions. Ventilation must always be considered as a removal process. For some pollutants and for some buildings, other removal processes can be important, such as deposition of particles onto indoor surfaces, irreversible reaction of a pollutant with an indoor surface, or active filtration.

Buildings are ventilated such that the replacement time of indoor air with outdoor air occurs on a time scale that is typically a few hours but may range from about 5 min in the case of a mechanically ventilated building using an economizer or a building with open doors and windows to about 10 h for a closed building that is on the tight end of the normal range. Dynamic, time-dependent relationships governing the indoor levels of outdoor pollutants are important for time scales similar to or shorter than the ventilation time scale, but the time-dependent processes are not so important for evaluating longer-term average conditions. In this letter, outdoor air pollutant impacts on indoor levels are assessed based on time-averaged conditions, rather than emphasizing short-term dynamics. On the other hand, short-term dynamics can be key in cases when high exposure concentrations lead to acute and severe health effects, such as CO poisoning.

2.6. Principles to ensure good indoor air quality

Key elements that help ensure good indoor air quality are these: first, avoid high indoor sources strengths for all pollutants that might be emitted indoors; second, ensure adequate ventilation; and third, use air filtration or air cleaning where needed to achieve further improvements. For many pollutants, the variability from one building to another in indoor emission rates has a more significant influence on concentrations than does variability in ventilation rates. High emission rates tend to produce poor indoor air quality irrespective of ventilation characteristics. On the other hand, even in buildings where the indoor emission rates are not excessive, low ventilation rates can produce indoor air quality problems. Ventilation efficiency also is important to consider (Sandberg 1981). The central principle is to remove pollutants where they are more highly concentrated and to supply clean air where people need it. The use of exhaust fans in bathrooms and range hoods above cooking appliances represent practical illustrations of efficient ventilation. Deliberate air cleaning for indoor environments is only widely practiced in the case of particle filtration in mechanically ventilated buildings. Opportunities exist to do more.

Starting from this framework, the following sections of this paper discuss how indoor air pollutant levels might be influenced by climate change. The discussion is organized according to pollutant source category and pollutant class, first considering indoor emission sources and second pollutants of outdoor origin. The treatment is not intended to be comprehensive, but rather broadly illustrative of important indoor air quality concerns that might be influenced by climate change.

3. Pollutants with important indoor sources

This section reviews important indoor air quality issues that are associated with indoor pollutant sources, and explores

how climate change might impact these issues. The emphasis is on conditions in the United States but the discussion is relevant for other countries with similar levels of economic development and similar buildings.

3.1. Pollutants from indoor combustion

Arguably, combustion is the most important source of air pollution. Indoor combustion for cooking, lighting, and heating has both a long history and a diverse prevalence of contributing to air pollution exposure. Lopez *et al* (2006) ranked 'indoor air pollution from solid fuels' as one of the top ten leading causes of global mortality and disease. This ranking is based mainly on the use of biomass and coal as cooking fuels in rural parts of developing countries. Unvented or incompletely vented combustion also occurs to a substantial extent in economically developed countries. Such combustion has a demonstrable impact on indoor pollutant concentrations and exposures. Evidence associating these exposures with public health consequences ranges from suggestive to clear and compelling.

Exposures resulting from indoor combustion could be altered in the future in several ways associated with climate change. Influencing factors could include changing prevalence, frequency, or strength of indoor emissions and also changes in building ventilation conditions. The following paragraphs summarize some of the concerns and provide references to document the nature and significance of the current problems.

3.1.1. Accidental CO poisoning.
Carbon monoxide (CO) is produced from the incomplete combustion of a carbonaceous fuel. Inhaled CO forms carboxyhemoglobin (COHb) in the blood, whose presence interferes with transport and delivery of oxygen to tissues and organs. Excessive acute exposures result in illness and death. Chronic lower-level exposures may also have health consequences, but the available evidence on this aspect is weaker than for acute poisonings.

Carbon monoxide is regulated as a pollutant in ambient air. Mainly through strong improvements in automotive emissions control technology, urban air levels of carbon monoxide have become well controlled, such that almost every area of the United States satisfies the National Ambient Air Quality Standard (www.epa.gov/air/oaqps/greenbk/cnc.html).

Despite improvement in outdoor air levels, carbon monoxide remains an important pollutant. Over the past decades, hundreds of accidental and fatal acute CO poisonings have occurred annually in the United States (Cobb and Etzel 1991, Mott *et al* 2002, King and Bailey 2008) and in Europe (Braubach *et al* 2012). The US mortality rate has declined, most likely because of improvements in the control of motor vehicle emissions. Mott *et al* analyzed CO-associated mortality statistics and concluded that, 'if rates of unintentional CO-related deaths had remained at pre-1975 levels, an estimated additional 11 700 motor-vehicle-related CO poisoning deaths might have occurred by 1998'. Holmes and Russell (2004) remarked that the reduced accidental

mortality resulting from improvements in motor vehicle emission controls 'is not accounted for in EPA's recent reports on the benefits and costs of the (Clean Air Act), yet it dwarfs the estimated direct benefits ascribed to CO control'.

In the context of climate change, a particular concern with carbon monoxide exposure arises in association with the use of portable electricity generators that burn liquid fuels such as gasoline. The utilization and reliability of centrally generated power might be degraded because of climate change for several reasons. For example, a shift away from fossil fuels toward wind and solar electricity might be accompanied by more frequent periods of service interruption because of the intermittency of renewable power sources. Hotter summer afternoons may precipitate more intense use of air conditioners, increasing the frequency of service demand overloads that cause brownouts and blackouts. Severe storm events can cause electricity service disruption. In any of these cases, people may rely more on their own electricity generators. If the generators are used indoors, or even when used outdoors but in too close proximity to an indoor environment, unhealthful carbon monoxide exposures can result. Increases in emergency room and other hospital visits caused by CO poisoning have been reported in association with power outages (264 individuals from 155 households; Muscatiello *et al* 2010), major storms (167 cases from 51 incidents; Van Sickle *et al* 2007), and floods (33 cases from 18 incidents; Daley *et al* 2001). It is reasonable to believe that the prevalence of CO-induced illness is larger than that recorded in the emergency-room statistics because illnesses which are not too severe may not be reported.

3.1.2. Cooking.
In the United States and other developed countries, cooking causes air pollutant exposures that have potential public health significance. For example, the use of natural gas as a cooking fuel is associated with elevated indoor exposures to nitrogen dioxide, a byproduct of the combustion process (Marbury *et al* 1988, Spengler *et al* 1994). An international epidemiological study in Europe reported associations between respiratory symptoms in women and gas cooking (Jarvis *et al* 1998). Exposure of children to elevated indoor nitrogen dioxide levels has also been reported to be associated with respiratory symptoms (such as wheeze) but not pulmonary function (Neas *et al* 1991). In a population of infants at risk for developing asthma, 'the frequency of reported respiratory symptoms in the first year of life was associated with NO_2 levels not currently considered to be harmful' (van Strien *et al* 2004). However, another study did not find an association between nitrogen dioxide levels and respiratory illnesses in infants (Samet *et al* 1993). It has been suggested that failing to account for nitrous acid (HONO) in such studies may be a contributing factor to inconsistency in the results (Jarvis *et al* 2005).

Exposures to ultrafine particles can be substantially increased by emissions from cooking (Bhangar *et al* 2011, Mullen *et al* 2011). Emissions of ultrafine particles are caused not only by combustion of the cooking fuel but also may result from high temperatures associated with electric cooking elements (Wallace *et al* 2008). Cooking is also

a significant source of indoor particle mass concentrations (Wallace *et al* 2004, Buonanno *et al* 2009). Fumes from Chinese-style cooking with hot oil have been shown to be mutagenic (Chiang *et al* 1997) and this cooking style has also been reported to be a risk factor for lung cancer in nonsmoking women in Taiwan (Ko *et al* 1997).

Climate change could impact the indoor concentrations of cooking associated pollutants in the United States and other developed countries in at least two ways. First, it may be that a mitigation response to climate change drives a movement toward smaller per-capita housing space (with lower life-cycle environmental impacts) and with lower air-exchange rates (to save heating and cooling energy). If so, then the emissions from cooking would be diluted into smaller volumes and would persist for longer times, which would tend to increase concentrations and exposures for a given level of cooking. Second, climate-change mitigation goals might push cooking away from the use of natural gas and toward a heavier reliance on electricity (Williams *et al* 2012). Such a shift would reduce associated exposures for NO_2 and for the ultrafine particles formed in combustion flames.

3.1.3. Space heating.   Combustion for space heating is sometimes associated with substantial pollutant emissions, especially because of the relatively large amounts of fuel used for home heating as compared, for example, to cooking. When on-site combustion is used to generate the heat, it is usually (but not always) the case that the heat is extracted from the combustion gases and the byproducts are vented outside. However, leakage may occur such that some of the generated pollutants enter the occupied indoor space of the same building for which the heat is being generated. In addition, combustion may also be unvented by design in some cases and then all of the byproducts formed are emitted into the indoor environment along with the generated heat.

Direct evidence that links household heating with health effects is limited. Household use of kerosene heaters and fireplaces for heating was found to be associated with respiratory symptoms in nonsmoking women in Connecticut and Virginia during the 1990s (Triche *et al* 2005). Unvented combustion heating (and cooking indoors with charcoal) was associated with carbon monoxide deaths in a study of coroners' reports in California (Liu *et al* 2000).

Climate change could induce several shifts that would affect indoor air pollutant exposures associated with heating. First, if average temperatures rise as expected, less heating may be needed and—all else being equal—there would then tend to be less associated pollution exposure. Climate-change mitigation efforts may also lead to better insulation of buildings, which would further reduce heating requirements. Second, there could be shifts in the types of heating sources used. Mitigation efforts could serve as a driving force for substituting electricity (from low-carbon sources) in place of fossil-fuel combustion, a change that would tend to improve indoor air quality. On the other hand, mitigation goals might also encourage greater use of renewably grown wood as a household heating fuel (Haluza *et al* 2012). As practiced today, residential wood combustion is

associated with degraded neighborhood air quality owing to emissions exhausted from the chimney, and also with degraded indoor air quality in the households that burn the wood owing to leakage of combustion byproducts into the indoor environment (Traynor *et al* 1987, Gustafson *et al* 2008). Increased wood-based heating could exacerbate indoor air quality problems associated with residential wood combustion.

Another potential trend that would tend to degrade indoor air quality is greater reliance on unvented combustion-based space heaters. Devices of this type have a high thermal efficiency because all of the generated heat is discharged indoors. However, their use can cause substantially elevated indoor concentrations of nitrogen dioxide, sulfur dioxide, and particulate matter (Leaderer 1982, Leaderer *et al* 1990, Francisco *et al* 2010). Also of concern is the inappropriate use of gas cooking appliances for space heating, a practice that has been implicated as a cause of accidental carbon monoxide poisonings (Liu *et al* 2000).

One additional concern associated with climate change and home heating is related to building envelope tightness. Efforts to save energy by reducing the leakiness of building envelopes can increase the risk of 'backdrafting', in which air flows into a building through the exhaust flue carrying with it combustion byproducts. The causes and consequences of backdrafting have received some attention in the literature (Nagda *et al* 1996), but the prevalence even for current conditions in the building stock has not been well characterized.

3.1.4. Smoking.   Habitual indoor smoking adversely affects indoor air quality and public health. Smoking indoors has a strong influence on indoor levels of fine particulate matter (Nazaroff and Klepeis 2004, Hyland *et al* 2008). Environmental tobacco smoke is also an important cause of environmental exposure to certain hazardous air pollutants, including acrylonitrile, 1,3-butadiene, acetaldehyde, acrolein, and formaldehyde (Nazaroff and Singer 2004). Evidence indicates that several severe adverse health impacts are associated with ETS exposure, including acute myocardial infarction (Lightwood and Glantz 2009), lung cancer (Fontham *et al* 1994), and a host of respiratory health problems in children (DiFranza *et al* 2004). Over the past few decades, there has been a marked reduction in exposure to ETS in the US population as reflected in lower concentrations of serum cotinine in nonsmokers (Pirkle *et al* 2006). The decline mainly reflects changes in the prevalence of smoking indoors, rather than changes in the building stock. Among other factors, the prevalence of smoking among US adults has declined from 37.4% in 1970 to 25.5% in 1990 and to 19.3% in 2010 (www.cdc.gov/tobacco/data_statistics/tables/trends/cig_smoking/index.htm). A national goal has been established to reach 12% smoking prevalence by 2020 (www.healthypeople.gov/2020/topicsobjectives2020).

In a future influenced by climate change, exposures to environmental tobacco smoke will be determined to a significant degree by the prevalence and intensity of smoking in indoor spaces occupied by nonsmokers. In the US, smoking

in public places has become uncommon. However, smoking in private residences continues. Singh *et al* (2010) have estimated that 7.6% of children in the US are exposed to ETS in their own home. Exposures to ETS occur not only within the residence in which smoking occurs, but also, in the case of multifamily dwellings, in neighboring units (Bohac *et al* 2011).

Changes in the residential building stock that are a consequence of climate-change concerns could influence exposure to ETS. Reduced household volume per occupant, lower air-exchange rates, and a shift toward more multifamily units might be consequences of efforts to mitigate anthropogenic impacts on climate. For a given source term (e.g., a fixed number of cigarettes smoked indoors per day), any of these changes would tend to increase exposures to ETS indoors.

3.2. Radon and its decay products

Indoor radon is a dominant cause of the public's health-relevant radiation exposure. Elevated residential radon exposure is an important risk factor for lung cancer. Based on a combined analysis of 13 studies collectively involving 7148 lung cancer cases and 14 208 controls, Darby *et al* (2005) concluded that residential radon is 'responsible for about 2% of all deaths from cancer in Europe'. In a parallel North American effort encompassing seven studies that collectively assessed 3662 cases and 4966 controls, Krewski *et al* (2005) found 'direct evidence of an association between residential radon and lung cancer risk, a finding predicted using miner data and consistent with results from animal and *in vitro* studies'.

Radon-222 (radon), the most health significant of the three naturally occurring isotopes, is generated by the radioactive decay of radium-226, a ubiquitous trace element in the earth's crust. Being an inert gas, radon can migrate from its parent material during its short lifetime (half-life = 3.8 days) and enter indoor or outdoor air where humans may encounter it. Radon does not directly pose a significant health hazard. However, its radioactive decay marks the beginning of a sequence of short-lived products. These radon decay products—isotopes of bismuth, lead, and polonium—are chemically reactive and so, when inhaled, can be retained on respiratory tract tissues. Subsequent radioactive decays irradiate lung cells. Of particular health concern are the alpha-particle emissions from the decays of polonium-218 and polonium-214. The radiation damage caused by these alpha-particle emissions creates the lung cancer risk associated with exposure to residential radon. The epidemiological evidence is consistent with a linear no-threshold dose–response model. Health risks from a given level of radon exposure are much higher for smokers than for nonsmokers, in accordance with smokers having a much higher underlying lung cancer risk. For example, Darby *et al* (2005) indicated that, in the absence of other causes of death, the absolute risk of lung cancer by age 75 for lifelong nonsmokers would rise from 0.4% for no radon exposure to 0.7% for chronic exposure to an elevated level of 400 Bq m^{-3};

for cigarette smokers, the corresponding risks would be 10% and 16%, respectively.

The three main sources of indoor radon are (i) soil near the building's foundation; (ii) earthen building materials, such as concrete; and (iii) tap water if taken from underground sources. Across the entire US building stock, soil is the most important radon source, although the other two sources dominate in some buildings. The significance of soil as a source of indoor radon depends on the radium content of the soil, on the permeability of the soil, and on the degree of coupling of the indoor space of the building to pore air in the underlying and adjacent soil (Nazaroff 1992). The only important removal mechanism of radon from indoor air is ventilation. However, the effective radiation dose to lung tissue associated with a given level of indoor radon depends on the dynamic behavior of the short-lived decay products (Porstendörfer 1994), which can be influenced not only by the ventilation rate, but also by factors such as indoor particle levels, active air filtration, and the intensity of indoor air movement.

Annual average residential radon levels in the United States have been estimated to have an arithmetic mean of 46 ± 4 Bq m^{-3} with an estimated 6% of dwellings exceeding the USEPA mitigation level of 148 Bq m^{-3} (Marcinowski *et al* 1994). The USEPA has estimated that 20 000 US lung cancer deaths annually are radon related (Pawel and Puskin 2004). In principle, control systems are well established for maintaining low indoor radon levels (Rahman and Tracy 2009). However, challenges remain to identify buildings with elevated concentrations and to apply effective and appropriate controls both in existing and new buildings.

Climate change might induce shifts in indoor radon and decay product concentrations for several reasons, although the direction and scale of the changes are difficult to predict. Changes that would reduce ventilation rates would tend to increase indoor radon levels and might also affect the radiation dose received per unit of radon exposure. Constructing buildings with materials that have high radium content should be avoided irrespective of climate-change concerns. The goal of improving the energy performance of buildings might induce increased use of subterranean spaces for habitation or stronger thermal coupling of building interiors to climate-buffered underground zones. Care would be needed in these cases to prevent radon levels from becoming elevated in the occupied spaces.

3.3. Organic compounds: VOCs and SVOCs

Organic compounds have been the subjects of many indoor air quality investigations. Exposure to volatile organic compounds (VOCs) has been associated with a variety of health effects, including allergic symptoms, asthma, and symptoms of sick building syndrome (Ten Brinke *et al* 1998, Smedje *et al* 1997, Norbäck *et al* 2000, Garrett *et al* 1999). Establishing definitive links between exposure to these compounds and specific health effects is challenging given that the amount of exposure sustained by study subjects and the conditions under which that exposure occurs generally are

beyond the direct control of the investigator. Also, human populations are exposed to multiple contaminants whose individual, let alone joint, effects are not known (Cohen and Gordis 1993).

For the discussion here, volatile organic compounds (VOCs) are those species that possess high enough vapor pressures to substantially volatilize and, when unconfined indoors, to be found primarily in the gas phase. Semivolatile organic compounds (SVOCs) are preferentially found in a condensed phase, but they still have sufficient volatility to have a meaningful vapor-phase presence. One classification system distinguishes SVOCs from VOCs on the basis of saturation vapor pressure. For SVOCs, the vapor pressure is in the approximate range 10^{-9} to 10 Pa, and VOCs have higher vapor pressures than 10 Pa. Alternatively, organic compounds can be classified as SVOCs or VOCs based on their boiling points.

Research on indoor VOCs began in the 1980s. Research on indoor SVOCs is much less developed. However, although there is not yet wide recognition of SVOCs as a category of indoor air contaminants, many studies have been reported on subcategories, including pesticides, polybrominated diphenyl ethers (flame retardants), and phthalates (plasticizers). Organic compounds with extremely low volatilities can also be present purely in the condensed phase and could still contribute to indoor air quality concerns as constituents of particulate matter. An important example of this category would be polycyclic aromatic hydrocarbons with many rings.

Several reviews have summarized the occurrence and potential health significance of VOCs and SVOCs indoors (Brown *et al* 1994, Wolkoff 1995, Wolkoff *et al* 1997, Jones 1999, Mendell 2007, Weschler and Nazaroff 2008, Rudel and Perovich 2009, Salthammer and Bahadir 2009). Logue *et al* (2011) compared published concentration data for US houses with health-based exposure guidelines and standards. Through this process, they identified seven specific VOCs as 'priority hazards based on the robustness of measured concentration data and the fraction of residences that appear to be impacted'. In alphabetical order, these are acetaldehyde, acrolein, benzene, 1,3-butadiene, 1,4-dichlorobenzene, formaldehyde, and naphthalene. Focusing on sensory irritation and other perceived indoor air quality impacts, Wolkoff *et al* (1997) have called attention to the importance of secondary pollutant formation indoors owing to reactive chemistry involving organic compounds and oxidizing agents such as ozone and nitrogen dioxide. Mendell (2007) reviewed 21 studies from the 'epidemiologic literature on associations between indoor residential chemical emissions, or emission-related materials or activities, and respiratory health or allergy in infants or children'. He found that the most frequently identified risk factors included 'formaldehyde or particleboard, phthalates or plastic materials, and recent painting'.

Emissions of VOCs tend to be higher following new construction and renovation activities, because of the release of chemicals from finite-capacity reservoirs in wood-based products, paints, floor finishes, glues and other construction and finishing materials (Dales *et al* 2008, Herbarth and Matysik 2010). House dust is a repository for semivolatile organic compounds and other particle-bound contaminants (Butte and Heinzow 2002). Results of studies on house dust have demonstrated the presence of polychlorinated biphenyls (PCB), polycyclic aromatic hydrocarbons (PAH), plasticizers (phthalates and phenols), flame retardants, other organic xenobiotics and inorganic constituents (Weschler and Nazaroff 2010).

Increasingly, biomarkers are being used to measure body burdens of environmental chemicals or chemical byproducts in human tissue (Paustenbach and Galbraith 2006, Sexton *et al* 2006). Recent work has emphasized the prenatal interval and infancy as highly vulnerable periods of development. Investigations have monitored indoor exposures to multiple chemicals and birth outcomes (Eskenazi *et al* 1999, Perera *et al* 2003, Rosas and Eskenazi 2008, Herbstman *et al* 2010). However, there is still a limited understanding of the relationships among environmental concentrations, biomarkers, and health outcomes.

With respect to the influence of climate change now and over the coming decades on exposures and public health risks from VOCs and SVOCs, perhaps all one can conclude is that the concerns are substantial enough to warrant further attention. In his review of indoor pollutants over the past fifty years, Weschler (2009) made an important point, stating, 'many of the chemicals presently found in indoor environments, as well as in the blood and urine of occupants, were not present 50 years ago. Given the public's exposure to such species, there would be exceptional value in monitoring networks that provided cross-sectional and longitudinal information regarding pollutants found in representative buildings'.

3.4. Carbon dioxide

Indoor exposures to carbon dioxide are likely to rise as a consequence of climate change. The atmospheric background concentration of CO_2 is rising. The preindustrial level was 280 ppm; the current level is above 390 ppm and is continuing to rise at a rate of a few ppm per year (www.esrl.noaa.gov/gmd/ccgg/trends/). There is also evidence of a rural–urban gradient, such that levels of CO_2 in outdoor air are higher in urban environments than in rural areas (George *et al* 2007, Jacobson 2010). In the United States, 58% of the year 2000 population lived in an urbanized area with a population above 200 000 (www.fhwa.dot.gov/planning/census/cps2k.htm).

Carbon dioxide levels are substantially elevated in occupied buildings as compared to outdoors. Although unvented combustion contributes, the main indoor source of CO_2 is the exhaled breath of building occupants. The metabolic production of CO_2 by humans depends on diet and on activity level. A typical CO_2 generation rate for a sedentary adult is 0.31 l min^{-1} (ASHRAE 2010), which corresponds to 34 g h^{-1} ($P = 1$ atm, $T = 293$ K).

Carbon dioxide levels are commonly used to guide ventilation practice in occupied buildings. In doing so, CO_2 serves as a marker of human bioeffluents. Research shows that, 'maintaining a steady-state CO_2 concentration in a space no greater than about 700 ppm above outdoor air levels will

indicate that a substantial majority of visitors entering a space will be satisfied with respect to human bioeffluents (body odor)' (ASHRAE 2010).

In common practice today, building ventilation requires energy use. In mechanically ventilated buildings, fan power is required to move air through ducts. When temperature or humidity of the outdoor air are not suitable for establishing desired indoor air temperature and humidity, then energy may be used for conditioning the ventilation air. Mitigation efforts to reduce anthropogenic climate change may create pressure to reduce ventilation rates in buildings to reduce the associated energy use. Hence, one might reasonably expect indoor CO_2 levels to rise in a climate-change influenced future for two reasons: (a) increased baseline levels because of rising outdoor CO_2 levels, especially in cities; and (b) reduced ventilation rates in buildings as part of a mitigation strategy.

Carbon dioxide is an acid gas; and, at high levels, 'inhalation of CO_2 can produce physiological effects on the central nervous, respiratory, and the cardiovascular systems' (www.osha.gov/dts/sltc/methods/inorganic/id172/id172.html). Recognizing its potential for frank adverse health effects, OSHA maintains an occupational standard for CO_2, with a 'transitional limit' of 5000 ppm for the 8 h time-weighted average (www.cdc.gov/Niosh/pel88/124-38.html).

As a matter of sound public policy, occupational standards for pollutants are commonly set at much higher levels than would be appropriate for the general public. However, in the United States, there are no health-based guidelines or standards for CO_2 levels *per se* that would apply for the general public in all indoor environments. In Germany, an indoor air working group from governmental agencies 'recommends the following guide values, based on health and hygiene considerations: concentrations of indoor air carbon dioxide levels below 1000 ppm are regarded as harmless, those between 1000 and 2000 ppm as elevated and those above 2000 ppm as unacceptable' (Anon 2008).

Are there public health consequences associated with exposure to CO_2 indoors at the levels at which they might be anticipated to occur in a future influenced by climate change? The literature does not provide a clear answer, but does contain some important clues. Shendell *et al* (2004) studied the association between student absenteeism and classroom CO_2 levels in Washington and Idaho. They found that 45% of classrooms studied 'had short-term indoor CO_2 concentrations above 1000 ppm. They also found a statistically significant association between higher levels of the indoor–outdoor difference in CO_2 level and student absenteeism. Haverinen-Shaughnessy *et al* (2011) measured CO_2 levels in 100 classrooms, inferred that 87 had substandard ventilation rates, and found a positive association between the inferred ventilation rates (in the range 0.9–7.1 liters per second (ls^{-1}) per person) and student performance on standardized tests. Seppänen *et al* (1999) reviewed the literature on ventilation rates, CO_2 concentrations, and sick building syndrome (SBS) symptoms. They found that, 'about half of the carbon dioxide studies suggest that the risk of sick building syndrome symptoms continues to decrease with decreasing (indoor) carbon dioxide

concentrations below 800 ppm'. Evaluating data from a study of 100 US office buildings, Erdmann and Apte (2004) found 'statistically significant, dose-dependent associations ($P <$ 0.05) for combined mucous membrane, dry eyes, sore throat, nose/sinus congestion, sneeze, and wheeze symptoms' in relation to the difference between indoor and outdoor CO_2 levels.

These associations do not provide evidence that CO_2 itself is harmful to public health at the levels ordinarily encountered indoors. It may be that the adverse effects reported are a result of some other contaminant whose concentrations are correlated with the indoor CO_2 levels. On the other hand, because of the role of CO_2 not only as a product of metabolism but also as a biological trigger to induce breathing, it is conceivable that levels of CO_2 encountered indoors might have direct health consequences.

A recent study by (Satish *et al* 2012) is one of the first investigations to isolate CO_2 from other contaminants and to explore its consequences for human health and well being. In that study, subjects were exposed during 2.5 h sessions in a well-ventilated chamber to different, steady levels of CO_2 controlled by its release from cylinders of ultrapure gas. Three conditions were established, with CO_2 levels of 600, 1000, and 2500 ppm. The subjects performed a computer-based test of decision making during each session. 'Relative to 600 ppm, at 1000 ppm CO_2, moderate and statistically significant decrements occurred in six of nine scales of decision-making performance. At 2500 ppm, large and statistically significant reductions occurred in seven scales of decision-making performance · · · but performance on the focused activity scale increased'. These findings, if confirmed, could have important ramifications for ventilation practices in buildings, especially those that are densely occupied, such as schools.

4. Pollutants originating from outdoor sources

Outdoor air pollutants enter buildings along with ventilation air. Depending on the pollutant and on building conditions, the indoor proportion of the outdoor pollutant level ranges from zero (i.e., perfect sheltering) to one (i.e., no benefit from being indoors). Carbon monoxide is an example of a pollutant for which a building provides effectively no protection on a time-averaged basis. For other species, such as particle matter and ozone, buildings can provide some protection against outdoor pollution. Filters in the mechanical ventilation system of typical commercial buildings actively remove some portion of the particles from air that passes through them (Hanley *et al* 1994). Particles also deposit onto indoor surfaces passively, thereby reducing the indoor proportion of outdoor particles (Riley *et al* 2002). Ozone can be deliberately removed from ventilation air using activated carbon (Shair 1981, Bekö *et al* 2009). However, the use of activated carbon in building mechanical systems is not common. On the other hand, ozone reacts rapidly with indoor surfaces and with certain gas-phase chemicals, most notably NO and terpenes. The indoor level of ozone attributable to its presence outdoors is attenuated, commonly

to 20–70% of the outdoor level (Weschler 2000). However, ozone reactions indoors generate byproducts, including formaldehyde, acrolein, and ultrafine particles, that have potential adverse effects (Weschler 2006).

These two categories of air pollutants—particulate matter and ozone—currently receive the most attention in outdoor air pollution control policy. Urban atmospheres are furthest from compliance with air quality standards. Furthermore, the attributable adverse health effects—based on epidemiological studies—appear to be higher for particles and for ozone than for other pollutants. The indoor–outdoor relationships for particulate matter and for ozone have been fairly well studied. What could happen to these pollutants in ambient air in a climate-change regime is important to consider. In addition to PM and ozone, which are considered in detail below, other ambient pollutants merit brief discussion. Several of these additional cases are summarized at the end of this section.

4.1. Particulate matter of outdoor origin

Airborne particles are a complex pollutant class, with source attributes, atmospheric dynamics, and health consequences that vary with size and chemical composition. The USEPA maintains health-based National Ambient Air Quality Standards (NAAQS) for particulate matter (PM) in outdoor air. These standards are based on 24 h average and annual average mass concentrations of particles finer than 10 μm in diameter (PM_{10}) and finer than 2.5 μm in diameter ($PM_{2.5}$). Particles in the diameter range 2.5–10 μm are referred to here as coarse whereas particles smaller than 2.5 μm in diameter are termed fine. Emerging evidence suggests that inhalation exposure to ultrafine particles, i.e. those smaller than 0.1 μm in diameter, also poses health risks in a manner that would not be well characterized or controlled by the existing NAAQS (Sioutas *et al* 2005).

Particulate matter in outdoor air is strongly associated with adverse health outcomes. Following a comprehensive review Pope and Dockery (2006), concluded that 'the literature provides compelling evidence that continued reductions in exposure to combustion-related fine particulate air pollution as indicated by $PM_{2.5}$ will result in improvements in cardiopulmonary health'. The coarse particle fraction (Brunekreef and Forsberg 2005) and ultrafine particles (Oberdörster 2001) have also been associated with adverse health effects that can be different than those of the fine particle fraction.

Atmospheric particles are often classified as primary or secondary. Primary particulate matter is emitted directly in the particle phase from sources. Secondary particles are formed in the atmosphere from the conversion of gaseous precursors to condensed phase species. Broadly, coarse particles are mainly of primary origin and tend to be mechanically generated, for example, by abrasion. Soil dust, sea salt, and fragments of tires, roadways, and vehicle brakes are examples of primary particles that are mainly found in the coarse mode. Coarse mode particles are commonly removed fairly rapidly (on time scales of minutes to hours) from the atmosphere. Because of their short atmospheric lifetime, concentrations of coarse

mode particles can be spatially heterogeneous with elevated concentrations near emission sources.

The fine mode is commonly a mixture of primary and secondary particles. Much of the primary material results from combustion processes and consists of noncombustible impurities or products of incomplete combustion, such as soot. Important secondary contributions to fine particles are associated with emissions of gaseous ammonia (NH_3), nitrogen oxides (mainly NO), and sulfur dioxide (SO_2). Atmospheric oxidation processes convert the nitrogen oxides and sulfur dioxide to nitric and sulfuric acids, which then combine with ammonia to form salts that condense onto preexisting particles to contribute to the fine-particle mass concentration. Another important source of secondary particulate matter is the emission of volatile and semivolatile organic compounds. Atmospheric oxidation processes can cause the partitioning of these species to shift between the gas phase and the condensed phase. Secondary particle formation tends to occur over a broad scale, in part because the hours to days required for atmospheric transformation processes allows for substantial transport and dispersion from local sources.

The intermediate particle size range (approximately 0.1–2 μm diameter) is also known as the accumulation mode because of the relatively long atmospheric persistence associated with these particles. They are too big to diffuse and too small to settle and so they persist for many days in the atmosphere. The combined importance of secondary formation and the slow atmospheric removal processes means that fine-particle mass concentrations exhibit a higher degree of spatial homogeneity than do coarse particles.

Atmospheric ultrafine particles have important primary sources, mainly tailpipe emissions from internal combustion engines (Kittelson 1998). They are also formed through secondary processes in the atmosphere referred to as nucleation events (Kulmala *et al* 2004). Ultrafine particles have relatively short atmospheric lifetimes. Number concentrations of primary ultrafine particles exhibit high spatial heterogeneity with very high concentrations on and near heavily traveled roadways. Secondary ultrafine particles are formed on a regional scale and so show more spatial homogeneity. However, the secondary formation events occur as bursts and so the temporal variability associated with these ultrafine particles can be high.

The degree to which particles of outdoor origin are present indoors depends on three main factors: (a) particle size; (b) building ventilation rate; and (c) the presence and degree of effectiveness of any filters used for removing particles from ventilation or recirculated air. Consider each of these factors in turn. First, with respect to particle size, the accumulation mode has the highest ability to penetrate and persist indoors (Riley *et al* 2002, Nazaroff 2004, Bennett and Koutrakis 2006). Coarse particles have a more difficult time penetrating infiltration cracks in the building envelope (Liu and Nazaroff 2001) or penetrating fibrous filters in ventilation systems (Hanley *et al* 1994). Coarse particles also deposit more rapidly onto indoor surfaces (Thatcher *et al* 2002). Similarly, ultrafine particles penetrate infiltration cracks less

effectively, are filtered more easily, and deposit onto indoor surfaces more rapidly (Lai and Nazaroff 2000) than do accumulation mode particles. Using a material balance model along with empirical data on governing factors, Riley *et al* (2002) estimated that an urban residence with a typical ventilation configuration would have indoor proportion of outdoor particles of approximately 0.45 for particle number concentration (mainly ultrafine particles), approximately 0.8 for $PM_{2.5}$ mass concentration, but only about 0.2 for coarse-particle mass concentration. These results illustrate that buildings provide some protection from exposure to particles in outdoor air.

In the absence of active filtration, higher ventilation rates tend to produce higher indoor concentrations of outdoor particles. The reason: ventilation serves as the sole source introducing outdoor particles into indoor air but as only one of several removal mechanisms. Higher ventilation rates increase the source term proportionately, but the removal rates less than proportionately. Furthermore, higher rates of ventilation provided by open doors and windows tend to allow penetration with little or no attenuation as compared to the typically lower fractional penetration associated with infiltration through cracks or mechanical ventilation through filters.

The effectiveness of filtration in providing protection indoors against particles of outdoor origin depends both on filtration efficiency and also on the airflow configuration. Filters are commonly used in mechanical ventilation systems and their efficiency can vary widely, as classified by the 'minimum efficiency reporting value' (MERV) (ASHRAE 1999).

In addition to these considerations, the chemical composition of particles can influence the penetration and persistence of outdoor particles in indoor environments. Lunden *et al* (2003) have shown that, under wintertime conditions, aerosol ammonium nitrate levels can be greatly reduced indoors as compared to the outdoor levels. For warmer indoor conditions, ammonium nitrate has an enhanced tendency to dissociate to the constituent gases, ammonia and nitric acid, and the nitric acid is then rapidly scavenged by the chemically basic gypsum wallboard commonly found indoors.

Regarding vulnerable populations, the findings of Hystad *et al* (2009) should be noted. They reported that 'residences with low (economic) building values had higher infiltration efficiencies than other residences, which could lead to greater exposure gradients between low and high socioeconomic status individuals than previously identified using only ambient $PM_{2.5}$ concentrations'.

Considering indoor particles of outdoor origin, what might one expect for a future in which climate change occurs? This question is best addressed in two parts. (a) What is expected to happen with regard to outdoor particles? (b) How might the indoor proportion of outdoor particles shift because of changing building design and operation?

Regarding aspect (a), it is useful to consider the possibilities sorted into several different categories of outdoor particles. This approach is used in the discussion that follows. In the final section 4.1.7, aspect (b) is addressed.

4.1.1. Regulated particulate matter. Particles in outdoor air are subject to air pollution control regulations. Given the strong regulatory, public policy, and technology momentum, and given that many areas in the US are currently out of compliance with existing NAAQS for PM, one might expect some overall improvement over the coming decades with regard to ambient particles levels, at least for PM_{10} and $PM_{2.5}$ for which the regulatory machinery is the strongest.

4.1.2. Sulfate from coal-fired power plants. Important contributions to improved ambient particle levels could be achieved by reducing sulfur dioxide emissions from coal-fired power plants. A transformation in the direction of lower S emissions from coal combustion could be accelerated because of climate-change concerns, since coal-fired electricity in the US accounts for a substantial proportion of global anthropogenic emissions of fossil carbon to the atmosphere.

4.1.3. Tailpipe emissions from motor vehicles. A shift away from the use of petroleum as a transportation fuel would have important benefits for reducing ambient particle concentrations. Because of the close proximity between urban roadways and buildings, tailpipe emissions from vehicles have a higher effectiveness in causing indoor air pollutant exposure per unit mass emitted than do central-station power plants, which emit their pollutants from tall stacks, often on the edge of or remote to populous regions. As with coal-fired electricity, an effective response to climate change in the transportation sector might yield some co-benefits in reducing indoor exposure to particulate matter. For example, a shift from vehicles powered by internal combustion engines to plug-in hybrid vehicles, to electric vehicles, or to fuel-cell-powered vehicles could lead to a significant net reduction in outdoor particle levels near buildings and consequent improvements in indoor air quality.

4.1.4. Residential wood combustion. Climate-change concerns may encourage increased use of wood combustion and the burning of other contemporary-carbon fuels for home heating. Residential wood smoke is an important contributor to ambient particle levels in the winter in many communities (McDonald *et al* 2000, Naeher *et al* 2007). It is feasible that wood combustion technologies could be improved to the point where excessive emissions are limited if not avoided (Olsson and Kjallstrand 2006, Ward *et al* 2010). On the other hand, Haluza *et al* (2012) have documented the potential for adverse health consequences from expanded residential wood burning with uncontrolled emissions for the case of Upper Austria.

4.1.5. Wildfires. Climate change is expected to increase the frequency of wildfires. Higher ambient temperatures combined with episodes of drought could lead to periods with a higher tendency for forests to burn. Park *et al* (2007) concluded that biomass burning is 'an important contributor to US air quality degradation, which is likely to grow in the future'. Spracklen *et al* (2009) have estimated that the annual mean area burned in the western United

States will be about 50% larger in 2050 as compared with year 2000, owing to climate-change impacts. Over the same time frame, they predict increases in summertime aerosol concentrations over the western US of 40% for organic carbon and 20% for elemental carbon, with most of the change attributable to increased wildfire emissions. Since wood smoke particles are primarily in the fine mode, ordinary indoor environments, especially residences, do not provide much protection. However, Barn *et al* (2008) have shown in an experimental study that using a recirculating, high-efficiency filter indoors can provide some protection against exposure to wood smoke associated with forest fires.

4.1.6. Windblown dust. Another expected effect of climate change is increased prevalence of drought, both in time and space. It is also anticipated that water resources will become further strained, which may lead to various pressures that could increase the dryness of land surfaces, such as reduced irrigation of crops and declining reservoir or lake levels owing to increased water extractions or diversions of influent streams. These conditions would have a tendency to increase the emissions of windblown dust into the atmosphere. Results from several recent studies illustrate the nature of the concern. Chan *et al* (2008) reported that Asian dust storms were associated with an increased frequency of emergency visits for ischemic heart disease, cerebrovascular disease, and COPD. Kuo and Shen (2010) reported increased levels of indoor $PM_{2.5}$ and PM_{10} in an office building during a dust storm. Hefflin *et al* (1994) reported on very high PM_{10} levels (more than 1000 μg m^{-3}) during seasonal dust storms in southeastern Washington State. On the basis of investigating daily emergency-room visits, Hefflin *et al* concluded that, 'the naturally occurring PM_{10} in this setting has a small effect on the respiratory health of the population'. In contrast, Ostro *et al* (2000) studied daily mortality in relation to particulate air pollution in the Coachella Valley, California, where 'coarse particles of geologic origin are highly correlated with and comprise approximately 60% of PM_{10}, increasing to >90% during wind events'. Their results demonstrated 'associations between several measures of particulate matter and daily mortality in an environment in which particulate concentrations are dominated by the coarse fraction'. Malig and Ostro (2009) assessed mortality statistics from 15 California counties for 1999–2005 in relation to coarse particle monitoring data. They found 'evidence of an association between acute exposure to coarse particles and mortality', and that 'lower socioeconomic status groups may be more susceptible to its effects'.

4.1.7. Indoor proportion of outdoor particles. With regard to the indoor proportion of outdoor particles, future conditions might be substantially different than current conditions in the building stock. However, the body of evidence is weak for making predictions about the nature and scope of change to be expected. The basis is even weaker for specifically attributing a portion of whatever evolution occurs to climate change. What is known for US conditions can be summarized as follows. US residential buildings have tended to become

more airtight with time (Chan *et al* 2005), certainly reducing air leakage rates. Traditionally, open windows have made important contributions to residential ventilation, so simply having a tighter envelope does not necessarily translate to lower air-exchange rates. However, recent measurements of ventilation rates in new single-family dwellings in California indicate that low rates are common in that portion of the building stock: 67% of 108 homes monitored had ventilation rates lower than the California building code requirement of 0.35 air changes per hour (Offermann 2009). Lower air-exchange rates would tend to provide improved protection for building occupants against particles of outdoor origin. However, with lower air-exchange rates, concentrations of pollutants from indoor sources would tend to rise.

A transition is underway in the US housing stock toward more widespread use of mechanical systems to provide ventilation (Russell *et al* 2007, Offermann 2009, Sherman and Walker 2011). Mechanical systems that provide supply air can be equipped with filters to remove particles. Good filtration efficiency is possible at modest cost (Fisk *et al* 2002, Bekö *et al* 2008). However, there are concerns that used filters in ventilation supply systems contribute to degraded indoor air quality, for example, being a factor in the occurrence of sick building syndrome symptoms (Bekö 2009). A survey of mechanical ventilation system performance in residential buildings in The Netherlands reports frequent shortcomings, including 'insufficient ventilation rates, high noise levels, unclean systems and insufficient maintenance' (Balvers *et al* 2012). So, further technological innovation and system improvements might be necessary to achieve economical yet reliable, and durable high-performance mechanical ventilation systems in residences that provide good protection for occupants against particles of outdoor origin.

4.2. Ozone and its byproducts

Ozone is a secondary atmospheric pollutant, formed by photochemical reactions involving nitrogen oxides and volatile organic compounds. Ozone concentrations in urban air have declined slowly over time in urban areas of the United States, resisting relatively vigorous efforts at controlling precursor emissions. As health science information has improved, the air quality standard for ozone has become progressively more stringent. Over time, the background level of ozone in the clean troposphere has risen (Vingarzan 2004). Consequently, the gap between baseline ozone levels in the absence of anthropogenic precursor emissions and allowed concentrations under the NAAQS has narrowed.

Several modeling studies have explored the consequences of climate change for outdoor ozone concentrations. Bell *et al* (2007) estimated hourly concentrations for 50 eastern US cities in the 1990s and 2050s, taking account of the expected change in climatic conditions (using IPCC Scenario A2) but not any changes in anthropogenic precursor emissions. A key finding from that study is that, 'on average across the 50 cities, the summertime daily 1 h maximum (ozone level) increased 4.8 ppb, with the largest increase at 9.6 ppb'.

Tagaris *et al* (2009) reported on the results of a detailed modeling study for the United States, comparing outdoor PM$_{2.5}$ and ozone levels between 2001 and 2050. As with Bell *et al*, this study also did not account for changes in emission sources or population. Tagaris *et al* estimated that climate-change-induced shifts in PM$_{2.5}$ levels would cause roughly 4000 additional deaths per year as compared with 300 additional deaths per year caused by increasing ozone concentrations. In an earlier study, Tagaris *et al* (2007) reported on model predictions for regional concentrations of ozone and PM$_{2.5}$ over the whole US, incorporating not only the direct effects of climate change (using IPCC emissions scenario A1B), but also anticipated emissions reductions for year 2050. They estimated that emissions reductions would be more than 50% for nitrogen oxides and sulfur dioxide. They found that, 'impacts of global climate change alone on regional air quality are small compared to impacts from emission control-related reductions'. Overall, they predict a 20% decrease in the mean summer maximum daily 8 h ozone levels over the US. They also predict that mean annual PM$_{2.5}$ levels will be 23% lower on average. Racheria and Adams (2009) published an analogous study in which they concluded that, 'climate change, by itself, significantly worsens the severity and frequency of high O$_3$ events over most locations in the US, with relatively small changes in average O$_3$ air quality'.

Buildings offer some protection from ozone exposure because ozone irreversibly decomposes on indoor surfaces and also reacts with some gas-phase species that may be found indoors. However, some ozone that penetrates does persist. With common residual ozone levels indoors, and because people spend the majority of their time in buildings, most ozone exposure occurs indoors (Weschler 2006). New evidence from research on ozone-initiated chemistry raises a potentially important question: to what extent are the health risks that are ascribed to ozone exposure influenced by the coincident exposure to the products of ozone-initiated chemistry? That chemistry, which produces potentially health-relevant volatile byproducts, such as aldehydes and organic acids, occurs on indoor surfaces (Weschler 2004), on clothing (Coleman *et al* 2008), on hair (Pandrangi and Morrison 2008) and even on human skin (Wisthaler and Weschler 2010).

The distinction is important in the context of considering climate-change impacts on indoor air quality and health. Changes in building design and operation can be anticipated owing to development of new materials, resource limitations, changing economic conditions, and changing fashion among other considerations (Weschler 2009). Such changes might deliberately or inadvertently alter the indoor to outdoor relationship for ozone, e.g. through the introduction of active or passive controls (Shair 1981, Lee and Davidson 1999, Kunkel *et al* 2010, Cros *et al* 2012) or through lowering the mechanical ventilation rate during periods of elevated outdoor ozone to limit its introduction into buildings (Walker and Sherman 2013). Such changes could also deliberately or inadvertently alter the nature, degree, and significance of ozone-initiated indoor chemistry. These two considerations

overlap but are not coincident. Overall, if ambient ozone levels increase while ventilation rates decrease, the net effect on indoor ozone concentrations is uncertain, but one would expect higher indoor concentrations of the byproducts of ozone-initiated chemistry.

4.3. Pollen

Pollen levels in outdoor air might rise as a consequence of climate change (Reid and Gamble 2009). Allergic rhinitis is a common malady. Pollen exposure, among other environmental factors, also can influence asthma incidence (Gilmour *et al* 2006). Intact pollen grains are relatively large (a few 10s of μm in diameter). As such, they should neither effectively penetrate into nor persist in indoor air (Liu and Nazaroff 2001, Sippola and Nazaroff 2003, Nazaroff 2004). Neither should they penetrate past the head if inhaled (Yeh *et al* 1996). Consideration of these factors would suggest that buildings would provide good protection against whole pollen grains and also that the biological insult associated with exposure to whole grains should be concentrated in the extrathoracic regions (eyes, nose, throat). The tracking of pollen grains into buildings (e.g., on clothing) might constitute an IAQ exposure and health risk concern if the grains are later resuspended indoors. Furthermore (and perhaps more importantly), pollen grains can fracture, generating much smaller particles (0.5–3 μm diameter) (Suphioglu *et al* 1992, D'Amato *et al* 2007) that carry allergenic proteins. These smaller particles could penetrate both the building envelope and the upper respiratory tract.

4.4. Sulfur dioxide

Ambient SO$_2$ levels are primarily a result of coal combustion and originate from the presence of sulfur as a percent-scale impurity in coal. EPA data show that ~68% of US national atmospheric sulfur emissions in 2002 were from 'electricity generating units' and that the other important sources are 'industrial/commercial/residential fuels' (~16%) and 'industrial processes' (~8%) (www.epa.gov/air/emissions/basic.htm#dataloc). Ambient air quality standards for SO$_2$, as well as acid-rain legislation (i.e., in the 1990 Clean Air Act Amendments), have led to substantial reductions (~50%) of SO$_2$ emissions from power plants. The largest remaining emissions are from older power plants whose high emission rates continue to be allowed. New coal-fired power plants are required to have good emission controls for SO$_2$ that are achieved, for example, using flue-gas desulfurization. If the use of coal to provide electricity (and potentially for liquid fuels) continues into the future without regard for climate, then ambient SO$_2$ levels might rise. However, an alternative possibility is that—to the extent that coal use for energy continues—it will be done in a manner that exhibits improved emission controls such that SO$_2$ emissions would decline. Also, indoor environments provide some protection against SO$_2$ because, as an acid gas, it reacts on indoor surface materials (Biersteker *et al* 1965, Walsh *et al* 1977, Grøntoft and Raychaudhuri 2004).

4.5. Nitrogen oxides

Nitrogen oxides (mainly NO and NO_2) are emitted primarily as a result of combustion processes. To some extent, the presence of N in fuel (as in coal) leads to NO_x emissions. However, any high-temperature combustion process that uses air as the oxidizer can also produce NO_x emissions, with the N originating from N_2 in the combustion air. Important sources of NO_x in ambient air are mobile sources (both on road and off-road), fossil-fueled power plants (coal and natural gas), and other stationary combustion of (mainly) fossil fuels. For 2002, EPA national emission inventory data indicate that mobile sources were responsible for about 60% of emissions for the United States. Because NO_x is a precursor to ozone and other photochemical smog components, it has been and continues to be subjected to strong emission-control efforts, and continuing progress in reducing emissions can be anticipated for the near future. A high level of scrutiny and emission control is especially anticipated for diesel emissions, which are becoming progressively more important (Dallmann and Harley 2010). Field monitoring evidence supports expectations that the use of catalyzed diesel particle filters can cause an increase in the emissions of nitrogen dioxide (Dallmann *et al* 2012), a regulated respiratory irritant. However, less future reliance on fossil fuels in particular and combustion in general suggests that NO_x emissions may eventually decrease in a future climate-change regime. The indoor environment provides modest to moderate protection against NO_2 of outdoor origin (Quackenboss *et al* 1986).

4.6. Hazardous air pollutants

In the United States, approximately 190 hazardous air pollutants (HAPs) were defined under the 1990 Clean Air Act Amendments. In contrast to the criteria pollutants, HAPs are regulated only in terms of emission limits from major sources—there are no ambient concentration standards. Concentrations of these pollutants are not routinely monitored. However, summary appraisals have combined emissions data with dispersion modeling and risk factors to discern which pollutants and where the health risks from HAPs are highest. For example, in one study, median hazard ratios (average ambient concentration divided by a cancer benchmark value) were highest for 1,3-butadiene, formaldehyde, benzene, carbon tetrachloride, chromium, methyl chloride, and chloroform (Woodruff *et al* 1998). For chronic (noncancer) toxicity, the highest median hazard ratio in the study of Woodruff *et al* was for found to be associated with acrolein. The indoor proportion of outdoor pollutants has not been well studied for these pollutants, although for benzene and for the chlorinated organics it is reasonable to expect that indoor environments provide little or no protection from outdoor levels. Future trends in the outdoor concentrations of these pollutants in a climate-change regime are not clear, although the scrutiny that they are receiving as HAPs suggests that emissions might decline over time.

5. Conclusions

Conceptually, the nexus of climate change, indoor air quality, and public health is simple. A balance between sources and removal processes governs indoor air pollutant concentrations for any species in the air of any indoor space. Concentrations in combination with human occupancy govern exposures. Excessive exposures confer health risks to those exposed. Climate change can affect this system in numerous particular ways, many of which have been reviewed in this letter. For example, by causing an increase in the outdoor concentrations of certain pollutants at certain places and at certain times, indoor concentrations and associated exposures in buildings at those places and times would increase.

Perhaps more important than the direct shifts caused by climate change are the shifts that are mediated by human responses to climate change. For example, mitigation measures to reduce energy use in buildings might lead to systematically lower ventilation rates in buildings that would cause higher concentrations and exposures to pollutants emitted from indoor sources. An adaptation measure might be the increased use of air conditioning, which could exacerbate anthropogenic emissions of greenhouse gases and, if accompanied by reduced ventilation rates, increase the concentrations of pollutants emitted from indoor sources. Reactions to climate emergencies also pose certain public health risks, such as the potential for poisoning from exposure to carbon monoxide emitted from portable electricity generators.

Dissected into its component parts, the elements of this system are indeed relatively simple. However, the elements that influence important outcomes in this system are numerous and diverse. Furthermore, these elements are interconnected in a complex manner that includes feedback loops and also interweaves natural processes with technology, individual human behavior, and social systems. It is these systemic features, rather than the nature of individual elements themselves, that pose the greatest challenges for understanding and effectively addressing the impact of climate change on indoor air quality and public health.

Because our overall understanding of this system is, as yet, limited, this article has focused on the factors that influence the indoor concentrations of health-relevant pollutants and how the concentrations might shift as a consequence of climate change. Three classes of factors were identified as having important influence: pollutant attributes, building characteristics, and human behavior. The diversity of building types is important, since issues of concern and appropriate responses differ among single-family dwellings, multifamily apartment buildings, schools, health care facilities, offices, and so on. It is also important to recognize and account for the diversity in subpopulations, in part because of variability in the degrees of susceptibility among individuals and groups to the effects of indoor air pollutant exposure. Furthermore, it is important to take account of variability within populations in the knowledge and resources with which to take effective action in response to changing conditions. Actions taken by individuals

can profoundly influence indoor air quality in individual buildings. In this respect, the system of indoor air quality and public health in buildings exhibits similarities with the safety aspects of the transportation system. In both cases, there are public as well as individual interests in seeing that the system works well, and negligent or ill-informed behavior by individuals can cause serious harm.

Focusing on pollutants, indoor concentrations can be decomposed into contributions from indoor sources and from outdoor air. Combustion is a major source of both outdoor and indoor air pollution and arguably produces the most important indoor air pollutants with respect to health risks. Important combustion-related issues associated with indoor emissions include carbon monoxide exposures from portable generator use and indoor air quality problems associated with cooking, heating, and smoking. Other important pollutants associated primarily with indoor sources include radon and volatile and semivolatile organic compounds. Outdoors, the main pollutants of concern are particulate matter and ozone. Specific particulate-matter concerns that may be exacerbated by climate change include increases in smoke from wild land fires, pollen, and windblown dust.

One major concern for climate-change impacts on indoor air quality is associated with reduced ventilation rates. There are two driving forces for this concern. First, as a mitigation measure, efforts are gaining momentum to save energy in buildings. Energy is required to condition the temperature and humidity of ventilation air and so it is tempting to save energy by reducing the rate of ventilation of indoor spaces. Second, as temperatures rise during the warm parts of the year, there may be a progressive shift to greater reliance on air conditioning and less on cooling by means of open windows. The effect of ventilation on indoor air quality has multiple facets that operate in different directions, which means that one cannot be certain of the net effect. A lower building ventilation rate will tend to provide some enhanced protection against certain pollutants from outdoors, such as particulate matter. On the other hand, reduced ventilation rates tend to cause concentrations to increase for pollutants that originate primarily from indoor sources. Reducing ventilation rates does not necessarily mean that indoor air quality problems will become worse; however, neither is it safe to assume that there will be no problems associated with reducing ventilation rates.

A second major concern about the effect of climate change on indoor air quality and public health is associated with indoor emission sources. For many pollutants with indoor sources, it has been found that variability in emissions exceeds variability in ventilation rates as the primary determinant of whether or not indoor air pollutant levels are excessive. Put another way, when the indoor emission rates are high, ventilation within a normal range is unlikely to be sufficient to avoid a problem. There is no evidence clearly linking elevated indoor pollutant emission rates to climate change. However, there are several potential concerns that deserve attention, including carbon monoxide from portable generators, emissions from cooking, emissions from heating systems (including unvented combustion appliances,

backdrafting, and increased use of wood as a fuel), emissions from smoking, radon from intimate contact with earthen materials, and volatile and semivolatile organic compounds from various indoor sources. Special attention is needed to ensure that life-cycle impact assessments aimed at improving the environmental performance of buildings take proper account that emissions from indoor materials and from indoor activities can have a disproportionately large impact on indoor air quality and public health.

The third major concern for the impact of climate change on indoor air quality would be attributable to changes in outdoor air pollution levels. To date, the scientific literature on the effects of climate change on outdoor air quality has focused on criteria pollutants, especially $PM_{2.5}$ and ozone. In the United States, and in many other countries with developed economies, there are good regulatory and technological systems in place that strive to reduce emissions from anthropogenic sources. The associated momentum is expected to continue to yield improvements in reducing ambient pollutant concentrations that are clearly associated with anthropogenic sources. Greater concern might be warranted in developing countries, especially in their megacities (Gurjar *et al* 2008, Chan and Yao 2008). Concern would also apply in developed countries for pollutants whose emissions lie outside of the regulatory system, such as smoke from wildfires, pollen from weeds, and windblown dust. Levels of pollutants such as these might rise strongly as a consequence of climate change. If so, indoor environments will be used as imperfect shelters that could be improved with proper attention and with a commitment of appropriate resources.

Acknowledgments

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